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Review

Comparison of alternative methods for managing the residual of material recovery facilities using life cycle assessment

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ABSTRACT

Excessive waste generation caused by exponential growth in resource use for the production of consumer goods, electronics and packaging has placed a growing burden on waste management globally. In Australia, waste is currently generated at a rate of 43 million tonne per annum and has a projected growth rate of 4.5% per annum. Diminishing landfill capacity adds to the pressure faced by governments to consider alternative waste technologies put forward by industry. In Australia, residual waste from material recovery facilities is under consideration by energy and waste companies for alternative management by waste-to-energy. This waste is not feasible to be efficiently separated for further processing. In this study, the environmental performance of the material recovery facilities' residual waste based in Sydney, Australia, is assessed using a life cycle assessment that estimates the potential impacts of acidification, climate change, eutrophication and photochemical oxidation. A sensitivity analysis tests different waste fractions of MRF residual waste composition. The study found that landfill had the lowest greenhouse gas emissions regardless of whether credits offset electricity, and of the carbon accounting methods used to measure biogenic carbon dioxide. The results also found landfill to have the lowest acidifying emissions but found the waste-to-energy technologies performed better in minimizing euthrophying and photochemical oxidation emissions. Aggregated by normalization and weightings, landfilling was found to have the lowest single score. The study reported electricity generation potentials through thermal turbine, synthetic gas engine and landfill gas combustion, and found incineration to have highest electricity generation potential, followed by gasification-pyrolysis.

1. Introduction

Globally, average material use has increased from 5.0 t to 10.3 t per capita per annum between 1950 and 2010 due to population growth, industrialisation and an increase in socio-economic power (Schaffartzik et al., 2014). Following this global trend, Australia now generates a total of 2.5 t of waste per capita per annum from municipal, commercial and construction waste based on a six year growth projection at 4.5% from a baseline of 1.9 t in 2011 (Commonwealth of Australia, 2010; Australia Bureau Statistics, 2011). Landfill is the primary method used to manage waste in Australia; however, some Australian state and local governments have shown interest in introducing waste-to-energy technologies predicated on improved efficiency and environmental performance (Coote, 2017; Lazzaro, 2017). In 2017, the Victorian state government announced a $2-million Waste-to-Energy Infrastructure Fund to support the development of waste-to-energy technology (Victorian Goverment, 2017). A proposed waste-to-energy plant in regional Victoria, claiming to avoid 500,000 t of carbon dioxide (CO_2) emissions per year from avoided natural gas, is also being assessed (Lazzaro, 2017). In Western Australia (WA), there are three waste-to-energy plants under construction to manage residual waste from material recovery facilities (MRF) (Douglas, 2014; New Energy, 2014, 2016). One plant argues its reason to not pre-treat MRF residual waste prior to incineration is because the treatment process would be too energy intensive and expensive (Douglas, 2014). The other plants planned to manage waste in low-temperature gasification have not specified any treatment of MRF residual waste in their process descriptions (New Energy, 2014,2016). In New South Wales (NSW), there

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is particular interest concerning the environmental feasibility of the refuse derived fuel (RDF), a residual waste to be managed by waste-to-energy instead of landfill (Energy Australia, 2017). The RDF is material that is not feasible for further material reprocessing that does not have a consistent composition or source (Luger, 2017). It is estimated to be one third biomass, with remaining materials including plastic and other non-recyclables (Luger, 2017). The Mount Piper Power Station, a black-coal fired power plant outside Sydney, is under consideration for separate combustion of RDF waste-to-energy (Luger, 2017). Potentially, a grate or fluidised-bed incinerator would generate an estimated 1.1 MW h of electricity per tonne of RDF (Luger, 2017). Energy Australia claims that the RDF feedstock can be considered as a renewable energy source (Coote, 2017). Indeed, the Renewable Energy (Electricity) Act 2000 does allow biomass-based components of MSW to be considered renewable (Commonwealth of Australia, 2016b). Energy Australia (2017) states that the proposed waste-to-energy facility managing 100,000 t RDF from residual waste per year can avoid 60,000 t of net greenhouse gas (GHG) emissions whilst supporting the NSW Government's best practice waste-to-energy policy (Energy Australia, 2017). The RDF would be sourced from residual waste from material recovery facilities (MRF) in Sydney (Energy Australia, 2017) but literature has not specified if that waste would be subject to pre-treatment or pre-sorting. The three WA waste-to-energy plants have not proposed to treat MRF residual waste feedstock. A final decision on the Mount Piper waste-to-energy plant is expected in 2018 (Energy Australia, 2017).

A general material recovery facility (MRF) is a multi-input, multi-output system used to recover post-consumer waste for reprocessing into new materials. MRFs receive co-mingled municipal solid waste (MSW), then sorts valuable materials including glass, paper, plastics and aluminium into single streams (War on Waste - Episode, 2017, KS Environmental Group, 2015). A typical MRF rejects, on average, 7.8% of material input (e.g., residual waste) (Carre et al., 2013). The MRF's residual waste contains non-recyclable materials and recyclable materials not in a physical form to be extracted through the mechanical separation process (War on Waste - Episode, 2017). Other high value options for MRF residual waste are unlikely, due to the variety of materials and the absence of quality controls to meet end market specifications, therefore the material is not considered feasible for furthering processing.

This paper aims to assess the alternative waste management of the MRF residual waste in Sydney, Australia, using life cycle assessment (LCA). Landfill is assessed as the status quo waste management technology. Incineration and gasification-pyrolysis are assessed as the alternative thermal treatment technologies.

The degradation of biomass waste in landfill produces direct emissions to air of CO_2 and methane (CH_4) (IPCC, 2006), commonly known as landfill gas (LFG). Electricity produced through the combustion of captured LFG can be exported to the electricity grid. Waste material also degrades to leachate, processed in a wastewater treatment plant, through which additional emissions to water and air are formed. The carbon that is lost to leachate generally represents less than 1% of total stock (IPCC, 2006). In LCA, this carbon is inventoried as total organic carbon (TOC), dissolved organic carbon (DOC), biological oxygen demand (BOD) and chemical oxygen demand (COD) (Doka, 2009). Other emissions from leachate include sulfate, hydrogen sulfide, ammonium, nitrate and phosphate (Doka, 2009). Incineration is the combustion of waste materials using a fuel, such as natural gas in a furnace to maintain temperatures of 800–1600 °C (Doka, 2003). The heat recovered from the furnace can drive a steam turbine to produce electricity. Incineration emissions include CO_2, carbon monoxide, water, sulfur oxides, nitrogen oxides, ammonia, hydrocarbons and organic acids (Gavrilescu, 2008). In addition, incineration produces slag, flue gas and wastewater (Doka, 2003). Ancillary materials products that contribute to indirect emissions include sodium hydroxide, calcium carbonate or lime, hydrochloric acid, iron chloride and ammonia (Doka, 2003). Gasification-pyrolysis has been reported to produce fewer air emissions than incineration (Khoo, 2009). The process uses two main thermal chambers; the first chamber is a reductive zone that compresses waste at 600 °C; and the second chamber is a high temperature gasifier, using oxygen and natural gas to reach temperatures of approximately 2000 °C (Hellweg, 2000). The melted inorganic residue from the gasifier forms a solid waste stream equivalent to slag in the incineration process. The synthetic gas (syngas) released from the gasifier chamber has a typical composition of hydrogen (25–42%), carbon monoxide (CO) (25 – 4%), CO_2 (10–25%) and water (Hesseling, 2002). It can be combusted in the gas engine to generate electricity. Ancillary materials products including sodium hydroxide, cement, iron chloride and general inorganic and organic chemicals contribute to process emissions on a system level, and include CO_2, CO, formaldehyde, non-methane volatile organic compounds and polychlorinated dibenzodioxins (Hellweg, 2000). Emissions to water include adsorbable organic halides (AOX) and COD compounds (Hellweg, 2000).

There are several waste management LCA studies, however, there is little analytical attention paid to the MRF residual waste. In fact, many LCAs of waste management including those in Victoria, Australia (Grant et al. (2001),(2003); and Carre et al. (2013)) focus on assessing the potential benefits of recycling over other waste management alternatives. An LCA study of recycling and landfill in NSW, Australia, assessed sensitivities of GHG emissions in landfill (Department of Environment Climate Change and Water NSW, 2010a). The study found the GHG impact of individual biomass materials resulted in a net decrease of 55% for paper, 41% for timber and 45% for garden waste if carbon sequestration was included. Finnveden et al. (2005) and Moberg et al. (2005) published a two-part LCA study which assessed the performance of waste packaging material (newsprint and PET) in landfill, incineration and recycling in Sweden. The carbon accounting methodology in these studies is similar to ours where carbon sequestration in landfill is not included and biogenic carbon emissions impacts are included. The study that has the most likeness in scope and concept with our study is by Assamoi and Lawryshyn (2012), who undertook an LCA of the residual waste of diverted MSW, in Toronto, Canada. The Assamoi and Lawryshyn (2012) study based the quantity of the MRF residual waste on projections from 2011 to 2040 to form the functional unit. The two scenarios developed were landfilling the entire functional unit, and incinerating 1000 t of waste per day whilst landfilling the remainder. The study's impact assessment uses characterisation factors for potential impacts of acidification, global warming and nutrient enrichment (Assamoi and Lawryshyn, 2012). The findings of Assamoi and Lawryshyn (2012) study showed that incineration outperformed landfill in each environmental impact category if credits for electricity produced from fossil fuels (coal, oil and natural gas) were included in the system boundary. Importantly, if these credits were excluded, then landfill outperformed incineration in each environmental impact category (Assamoi and Lawryshyn, 2012).

For the case of gasification-pyrolysis, there is also limited available LCA research. Zaman (2010) researched the treatment of MSW in Sweden through landfill, incineration and gasification-pyrolysis, similar to the scope of our study. However, the MSW is not broken down by material, nor are the emissions reported by sub-processes in the system, making origins of burdens untraceable. The study also included the energy inputs to operate the technologies and the energy generated on a system level, however, the energy in relation to MSW-heating values are not reported. The major findings were that for acidification and eutrophication, landfill had the lowest impact, while for global warming potential (equivalent to CCP in this study) gasification pyrolysis had lowest impact, and incineration has lowest photochemical oxidation impacts (Zaman, 2010). Another study by Zaman in 2013 of the treatment of MSW in incineration and gasification-pyrolysis found gasification-pyrolysis performed better in relation to acidification by approximately 58%, GWP by approximately 2%, eutrophication by approximately 35% and photochemical oxidation by approximately

60% (Zaman, 2013). A study in Singapore by Khoo (2009) analysed the environmental performance of eight different advanced thermal treatment technologies that produce syngas including gasification-pyrolysis. The results for GWP showed that the start-up energy was the largest contributor of GHG for all technologies and gasification-pyrolysis produced approximately 165 kg CO_2 –eq per tonne of syngas (Khoo, 2009).

This paper contributes to the body of knowledge of the environmental performance of waste management options for the MRF residual waste in Sydney, Australia. To address uncertainty in relation to the composition of waste fractions in MRF residual waste, three other waste composition sources based in Australia, United Kingdom and Canada have been analysed in a sensitivity analysis. Therefore, this study enables a deeper and more regionally-relevant understanding of the environmental profile of waste-to-energy technologies in Sydney.

This study is timely as there is an increasing trend to consider thermal waste treatments to manage waste in Australia (Douglas, 2014, New Energy, 2014, 2016, Energy Australia, 2017, Victorian Goverment, 2017). The results from this study are expected to trigger further stakeholder interest in the environmental credentials of different waste management technologies. These stakeholders include the NSW Government, other Australian state governments, the waste management industry, the energy industry and other researchers. The results could additionally serve as the basis for further research aimed at informing public policy and planning for alternative waste management in NSW.

2. Methodology

The study uses LCA as the basis for determining the environmental performance of the MRF residual waste in Sydney over one year. The four LCA phases used were:

1) Goal and scope definition, which defines the study's aims, intended use and processes to be studied (system boundary)
2) Life cycle inventory (LCI), comprising the compilation of environmental input resources and emissions associated with the processes in the system boundary
3) Life cycle impact assessment (LCIA), which translates the LCI into indicators of potential environmental impact
4) Interpretation phase, where the LCI and LCIA results are examined as a basis for discussion, conclusions and recommendations in relation to the goal and scope.

The detail of these LCA phases are explained in subsequent sections. SimaPro 8.0.4 was used for the life cycle inventory modelling and life cycle impact assessment (LCIA).

2.1. Goal and scope

The total generation of MSW kerbside recycling in Sydney metropolitan area was 369,704 tonnes in 2014/2015 (NSW EPA and State of NSW, 2016). Of that waste, approximately 7.8% (Carre et al., 2013) will form the MRF residual waste (i.e. 7.8% of 369,704 = 28,837 t). Therefore, the functional unit was defined as the management of the MRF residual waste from kerbside recycling MSW in Sydney over one year, estimated at 30,000 ts.

The system boundary in Fig. 1 displays the processes included in the study - the collection and transport of the MSW to the MRF, and the MRF energy inputs and transport for each waste treatment scenario. In addition, the waste treatment scenarios include the infrastructure, process burdens from waste treatment, energy inputs, ancillary materials, solid waste disposal, internal transport, direct emissions from waste processing and credits from avoided electricity production. The study has been conducted as a waste management LCA and therefore has excluded process stages of material extraction, material processing and consumer use. This approach is called a zero-burden assumption, meaning once the holder discards the waste product, the life cycle starts and all treatment process are included until the material ceases to be waste and becomes an emission to air, emission to water or an inert material in landfill (Buttol et al., 2007). As the recyclate component is not included in the study, all the impacts associated with the collection and transportation and those associated with the MRF are allocated entirely to the residual waste.

2.2. Life cycle inventory data

This section outlines the inventory data used in the LCA model: the composition of MRF residual waste by waste fractions including the baseline and sensitivity analysis, electricity grid from NSW, and the landfill, incineration and gasification-pyrolysis systems under study.

2.2.1. Residual waste composition

The composition of the MRF residual waste was based on waste survey results APC (2014), which analysed the MRF residual waste going to landfill from the Hume MRF in the Australian Capital Territory. This data, derived from APC (2014) excludes liquids (e.g. beverage containers that are not empty) and breaks down composite materials (e.g. electrical appliances) into single materials based on estimates of material compositions. The waste material composition was applied to the functional unit so that material-specific LCI could be used in each scenario and display an aggregate material flow. Waste Composition 1 was based on the previous report of the Hume MRF survey (APC, 2009), Waste Composition 2 was based on data reported in an LCA study of MRF residual waste in Toronto, Canada (Assamoi and Lawryshyn, 2012), and Waste Composition 3 was based on a MRF survey in the United Kingdom (Enviros Consulting, 2009). Table 1 displays the MRF residual waste in this study and those adapted from other literature for the sensitivity analysis.

2.2.2. Electricity grid mix

The NSW electricity production mix to the grid was based upon the production data for the 2013/2014 financial year (Commonwealth Of Australia et al., 2015), shown in Table 2. The Australasian Unit Process (Life Cycle Strategies, 2015) were used for the inventory data for each electricity production source. Additional data about electricity production and transmission was based on Swiss approximations to Australian processes from Australasian Unit Process (Life Cycle Strategies, 2015). The additional data accounted for 4.50×10^{-6} kg ozone per kWh delivered (low population), 5.00×10^{-6} kg of dinitrogen monoxide emissions per kWh delivered (low population) and 4.3% energy losses through transmission. In addition, the transmission network was 8.44×10^{-9} km per kWh delivered from Swiss approximations.

2.2.3. Shared processes

The transport is assumed to have diesel usage of 11.0 L per tonne of waste for a disposal distance of 17 km from the kerbside to the MRF and 20 km to each waste facility. Carre et al. (2013), based on models of waste collection, reported 11.0 L of diesel fuel per tonne of waste was required to transport kerbside recyclables to the MRF in metropolitan Melbourne and the same data has been used as an equivalent for metropolitan Sydney. The transport distance of the MRF residual waste to landfill was 20 km in 2010, based on avoided landfill estimates for kerbside recycling (Department of Environment Climate Change and Water NSW, 2010b). The transport distance of the MRF residual waste to either incineration or gasification-pyrolysis is assumed to be 20 km, the same as the distance from the MRF to landfill.

The inputs to operate the MRF were based on Carre et al. (2013), who accounted for electricity at 27.45 kW h per tonne, energy from LPG at 31.44 MJ per tonne and an infrastructure input at 1.84×10^{-8} plant (p) per tonne.

2.2.4. Landfill LCI

Material specific carbon stocks and degradability values, reported in

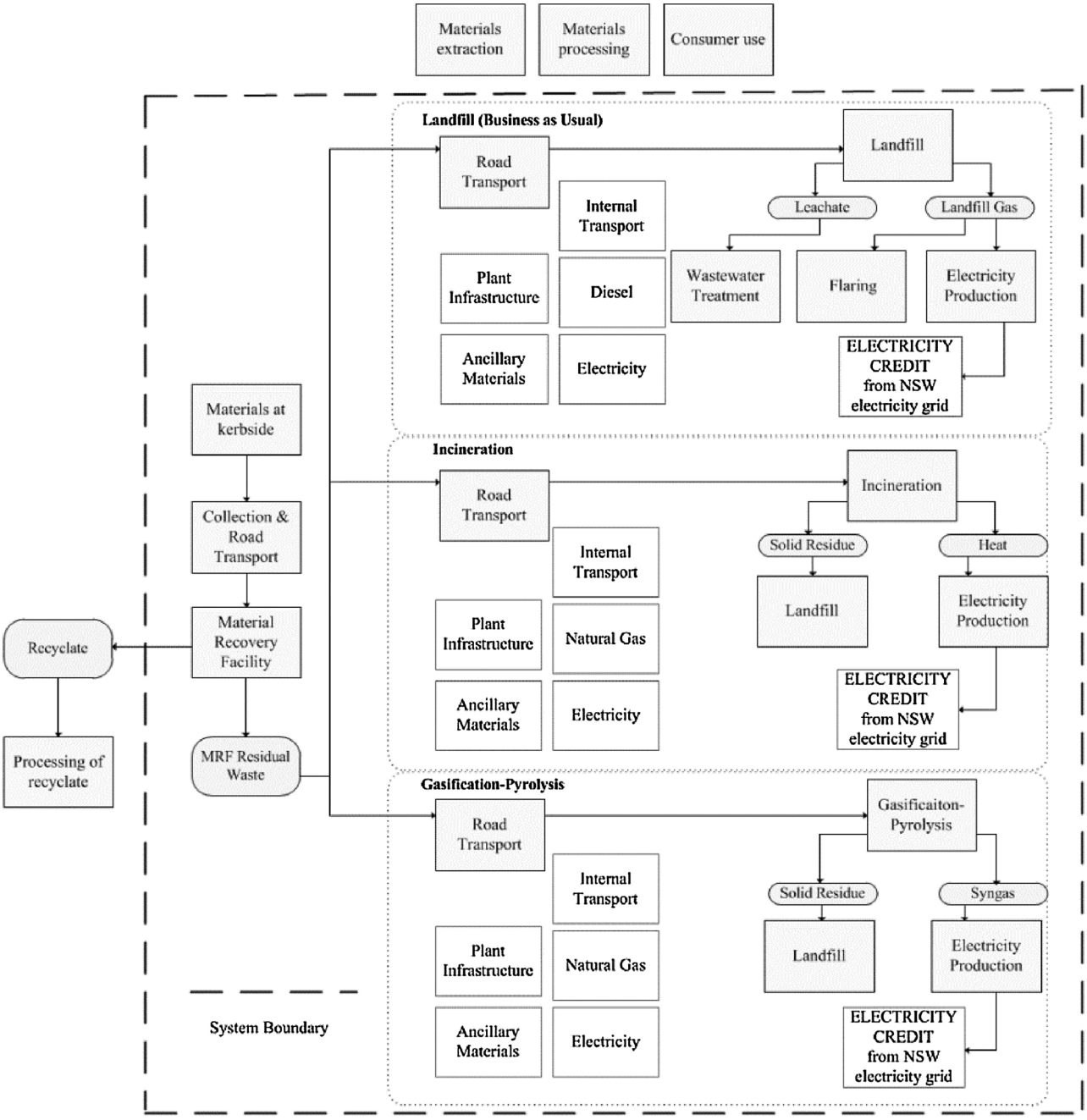

Fig. 1. System boundary diagram of the study. The included processes represent collection and transport from kerbside to MRF, and the MRF energy inputs and transport for each waste treatment scenarios including credits against avoided electricity production. The excluded processes represent material extraction, material processing, consumer use and recycling.

Table 3, were used to calculate CO_2 and CH_4 emissions using the IPCC first order decay (FOD) method (Commonwealth of Australia, 2017, Barlaz, 1998, IPCC, 2006). The method calculated the carbon emissions to soil using the initial carbon (C_i), the degradable organic carbon (DOC) and the non-degradable organic carbon fraction (DOC_f). The decay rate (k) was used to calculate the timing of CH_4 and CO_2 emission releases over 100 years. The values of C_i were sourced from Barlaz (1998) and DOC, DOC_f and k were sourced from Commonwealth of Australia (2017).

The direct emissions from leachate to water for the MRF residual waste were calculated based on the method in Doka (2009) using waste-specific degradability, elemental transfer and release factors. The emissions to air resulting from the treatment of leachate are CH_4 and dinitrogen monoxide, and were calculated using the IPCC method (IPCC, 2006). The calculations of COD (emissions to water) in the wastewater treatment from the anaerobic digester used the CH_4 air emissions, and assumed a CH_4 correction factor of 0.8 and an emission factor of 0.25 (IPCC, 2006). The emission factor used to calculate dinitrogen monoxide from nitrogen was assumed to be 0.005 (IPCC, 2006). The infrastructure for a landfill site of 1,800,000 m^3 capacity included inputs of clay, lining, crushed aggregate, polyethylene pipework for landfill gas capture, diesel, transportation and land occupation and was based on reported data in Carre et al. (2013). The plant burden was 5.56×10^{-7} p per tonne, assuming a waste input density of 1 t per m^3 (Carre et al., 2013). For the wastewater plant infrastructure, the plant burden was 6.67×10^{-12} p per kg of leachate, based on inventory in AusLCI Unit Processes (AusLCI, 2015). Over the time horizon of 100 years, the LFG collection for generation of electricity is assumed

Table 1
Fraction of waste material composition (mass %) in the MRF residual waste in metropolitan Sydney for this study and the compositions of MRF residual waste for sensitivity analysis.

Waste Material	Baseline	Waste Composition 1	Waste Composition 2	Waste Composition 3
Aluminium	1.6%	2.0%	0.7%	2.6%
Food	3.4%	12.9%	37.3%	–
Garden	1.2%	2.4%	8.6%	–
Glass	44.4%	14.1%	2.8%	6.5%
Hazardous	0.2%	0.8%	–	–
Masonry	0.7%	0.4%	–	–
Metals, general	2.1%	7.2%	2.1%	–
Nappies	1.0%	3.0%	–	–
Paper, mixed	15.0%	6.9%	26.1%	37.9%
Plastics, mixed	18.2%	38.0%	18.2%	46.8%
Steel	9.1%	6.6%	–	2.3%
Textiles, natural	2.0%	3.2%	0.6%	4.0%
Timber	1.2%	2.4%	3.5%	–

Table 2
New South Wales electricity production mixes from 2013/2014.

PRODUCTION SOURCE	Allocation
Black coal	81.1%
Natural gas	8.2%
Oil products	0.3%
Bagasse	0.7%
Biogas	0.6%
Wind	1.3%
Hydro	6.3%
Solar	1.6%

NB: Data source is from Commonwealth Of Australia et al. (2015).

Table 3
Material specific parameters for landfill degradation calculations using IPCC FOD method.

Waste Material	Ci	DOC	DOC_f	k
Paper mixed	0.403	0.4	0.49	0.06
Garden	0.4487	0.2	0.47	0.10
Nappies	0.502	0.24	0.50	0.04
Textiles	0.502	0.24	0.50	0.06
Timber	0.494	0.43	0.10	0.03
Food	0.508	0.15	0.84	0.18

for 30 years, flared for 10 years and not collected for the remaining 60 years (Pickin, 2010).

2.2.5. Incineration LCI

The incineration technology was predominately based on the ecoinvent 3.0 database, developed from literature on commercial MSW incinerators in Switzerland circa 2007, and regarded as suitable proxies for modern waste incineration (Doka, 2013). The technology mix within ecoinvent 3.0 is 24.5% selective non-catalytic reduction (SNCR), 42.8% selective catalytic reactor (SCR) high dust and 32.7% SCR low dust (Doka, 2013). Efficiency for electricity production was 15.8%, efficiency for heat recovery, 28.5%, and internal electricity usage was 0.13 kW h per kg waste, based on average MSW incineration in ecoinvent 3.0 (Doka, 2013). The natural gas input was based on an internal heat demand of 0.49 MJ per kg waste (Doka, 2013) and an assumed efficiency factor of 80% for heat losses in combustion, equalling 0.61 MJ per kg. This study's waste inventory has direct emissions to air and water from combustion of the waste from ecoinvent 3.0 for aluminium, steel, metals general, glass mixed paper, mixed plastic, nappies, masonry, timber and hazardous, and AusLCI Unit Processes for garden, food and textiles. Ancillary materials to the process, infrastructure, process specific burdens, internal transport and residual waste were based on background processes in AusLCI Unit Processes and Australasian Unit Processes (AusLCI, 2015, Life Cycle Strategies, 2015).

Table 4
Electricity energy credits (MJ per functional unit) under each waste treatment scenario.

	Landfill	Incineration	Gasification-Pyrolysis
Baseline	7.87×10^6	4.15×10^7	2.65×10^7
Waste Composition 1	8.61×10^6	7.42×10^7	4.04×10^7
Waste Composition 2	2.37×10^7	5.80×10^7	4.53×10^7
Waste Composition 3	1.57×10^7	9.65×10^7	5.87×10^7

2.2.6. Gasification-Pyrolysis LCI

The inventory for gasification-pyrolysis direct emissions were calculated from a mass balance using elemental transfer paths in Hellweg (2000). The elemental composition of individual waste materials', used in the mass balance calculations were from Doka (2003). From the mass balance, emissions from mineral residue (slag equivalent), syngas and other residues such as sulfur, heavy metals, salts and condensate were calculated. The production of electricity from syngas was estimated at 0.5 MJ per kg (Hellweg, 2000), and was used to calculate the electricity credits in Table 4. Process burdens are emissions that were not traceable to individual waste materials but are known on a system level and included additional CO_2, CO, dioxins, particles, AOX and COD were sourced as a priori from Hellweg (2000). The ancillary materials were based on Hellweg (2000), however, their quantities were based on the equivalent waste incineration inventories in ecoinvent 3.0, AusLCI Unit Processes and Australasian Unit Processes. The infrastructure inputs were also assumed to be the same as equivalent waste incineration inventories by AusLCI Unit Processes and Australasian Unit Processes.

2.2.7. Electricity from waste LCI

The electricity from MRF residual waste treated in landfill was calculated using a conversion rate of 50 MJ per kg CH_4, an efficiency factor for generation of 35% and the IPCC FOD method to quantify LFG (Carre et al., 2013, IPCC, 2006). The electricity from incineration was based on lower heating values (LHV) of waste materials and efficiency factor for generation of 15.8% (Doka, 2013). The electricity from gasification-pyrolysis was calculated using the LHV of syngas of 9.1 MJ per kg, the quantities of syngas calculated, a fraction syngas sent to gas engine of 93.4% and, an efficiency factor for generation of 32% (Hellweg, 2000). Table 4 shows the net electricity credits in landfill, incineration and gasification-pyrolysis from this study and the three sensitivity analyses.

3. Life cycle impact assessment

To assess the environmental performance, life cycle impacts were calculated across the following metrics - acidification potential (AP), climate change potential (CCP), eutrophication potential (EP) and photochemical oxidation potential (POP). The temporal scope was limited to a 100-year time horizon. For EP, AP and POP the CML baseline method was used (Institute of Environmental Sciences, 2016). CCP was assessed over a 100 year timeframe and used the GWPs taken from the IPCC's forth assessment report (IPCC et al., 2007). Also relating to CCP, carbon of biogenic origin was excluded from the baseline impact assessment model, but addressed with a sensitivity analysis that includes carbon sequestration. Table 5 shows the characterisation

Table 5
Characterisation factors for CCP for the biogenic CO_2 non-neutral and biogenic CO_2 neutral LCIA methods.

GHG Emission	Biogenic CO_2 non- neutral (kg CO_2 -eq) (A)	Biogenic CO_2 neutral (kg CO_2 –eq) (B)
CO_2 biogenic emitted	1	0
CO_2 biogenic sequestered	0	−1
CO_2 fossil emitted	1	1
CH_4 biogenic emitted	25	22.25
CH_4 fossil emitted	25	25

NB: (A) from IPCC et al. (2007);(B) from Muñoz et al. (2013).

factors for GHG emissions to determine CCP for the baseline and the sensitivity analysis. The baseline uses the biogenic CO_2 non-neutral method, based on the IPCC fourth assessment report (IPCC et al., 2007); and the sensitivity analysis uses the biogenic CO_2 neutral method, based on Muñoz et al. (2013).

The normalization of the characterized results was based on Australian government data estimating total national emissions for each indicator in 2015 (Commonwealth Of Australia, 2016b,a). Normalization factors were the inverse of total estimated emissions and are shown in Table 6.

Four weighting methods, shown in Table 7, were applied to the normalized results from each indicator to calculate a single score result. These weightings were distributed as equal, skewed towards non-CCP, Australian weightings based on Building Productions Innovation Council (Building Products Innovation Council, 2018) (Bengtsson et al., 2010) and international weighting based on BEES + Version 4.05 (Building For Environmental And Economic Sustainability et al., 2008).

Uncertainty analysis of the results was done by a comparative Monte Carlo (MC) simulation on SimaPro 8.0.4 using 1000 run iterations in a 95% confidence interval. Uncertainty factors for data inventory were estimated using a Pedigree matrix to qualitatively assess based on upon reliability, completeness, temporal correlation, geographical correlation, technological correlation and sample size. The uncertainty analysis compared landfill to incineration, landfill to gasification-pyrolysis and incineration to gasification-pyrolysis, for the treatment of 30,000 t of MRF residual waste.

4. Results

4.1. Characterized impacts

The results summary of the characterized impacts calculated in the LCA model for the treatment of 30,000 t of MRF residual waste is shown in Table 8.

4.2. Single score results

Single score results presented in Table 9 show that for the treatment of 30,000 t of MRF residual waste, landfill has the lowest score and gasification-pyrolysis has the highest single score result under each weighting method.

Table 6
Normalization factors in reference to total estimated emissions in Australia.

Indicator	Normalisation Factor
AP	4.35×10^{-10}
CCP	1.86×10^{-12}
EP	1.74×10^{-10}
POP	2.19×10^{-11}

Table 7
Weighted factors applied for each indicators' normalized impacts to calculate a single score result.

Weighting	AP	CCP	EP	POP
Equal	25%	25%	25%	25%
Skewed to non-CCP	30%	10%	30%	30%
BPIC	25%	30%	9.7%	15.6%
BEES+	25%	30%	9.7%	18.8%

Table 8
Results of characterized impacts for the treatment of 30,000 t of MRF residual waste in landfill, incineration and gasification-pyrolysis.

Indicator	Unit	Landfill	Incineration	Gas.-Pyrolysis
AP	kg SO_2 -eq	4.57×10^3	1.01×10^4	2.00×10^4
CCP	kg CO_2 -eq	1.07×10^7	1.55×10^7	2.30×10^7
EP	kg PO_4^{3-} eq	1.26×10^4	2.39×10^3	4.79×10^3
POP	kg C_2H_2-eq	1.83×10^3	3.95×10^2	4.39×10^2

Table 9
Singe score results of normalized impacts for the treatment of 30,000 t of MRF residual waste using four different weighting methods (GP = gasification-pyrolysis).

Weighting	Landfill	Incineration	GP
Equal	6.02×10^{-6}	8.44×10^{-6}	1.31×10^{-5}
Skewed to non-CCP	3.25×10^{-6}	4.34×10^{-6}	7.15×10^{-6}
BPIC	1.39×10^{-5}	2.02×10^{-5}	3.02×10^{-5}
BEES+	1.06×10^{-5}	1.52×10^{-5}	2.29×10^{-5}

4.3. Process drivers of characterized impacts

These results show the process drivers of the characterized impacts based on the functional unit. A negative result indicates a net environmental benefit (i.e., credit from the electricity grid), whilst a positive result indicates a net environmental burden. The results for each indicator shown in Fig. 2 (AP), Fig. 3 (CCP), Fig. 4 (EP) and Fig. 5 (POP) include the process drivers contributing the net result. The black diamond represents the net outcome, the processes that are positive represent environmental burdens and the processes that are negative represent environment benefits from avoided processes.

Gasification-pyrolysis has the highest AP at 1.99×10^4 kg SO_2 –eq, 76% greater than incineration and 325% greater than landfill. The combustion of natural gas emitting nitrogen oxides is the primary driver of AP, followed by electricity used in the process, also emitting nitrogen oxides.

Table 10 presents the elementary flows of acidifying substances contributing to AP for each system. In landfill, nitrogen oxides contribute to 76% and sulfur dioxide (SO_2) contributes to 23% of the total impact. Nitrogen oxide emissions are mostly emitted from diesel combustion in transport, diesel combustion in landfill operations and the combustion of LFG. In incineration, nitrogen oxides contribute to 84% and SO_2 contributes to 11% of the total impact. Nitrogen oxide emissions are mostly from the production of electricity to the NSW grid, natural gas for combustion and the emissions from combusted waste. In gasification-pyrolysis, nitrogen oxides contribute to 89% and SO_2 contributes to 10% of the total impact. Nitrogen dioxide emissions are mostly from the production of electricity to the NSW grid, natural gas for combustion and the emissions from combusted waste.

Gasification-pyrolysis has the highest CCP at 2.30×10^7 kg CO_2 –eq, 46% greater than incineration and 115% greater than landfill. Direct emissions from the combustion of waste contribute to biogenic CO_2 and fossil CO_2. The combustion of natural gas for maintaining high temperature in the gasification chamber contributes to additional fossil

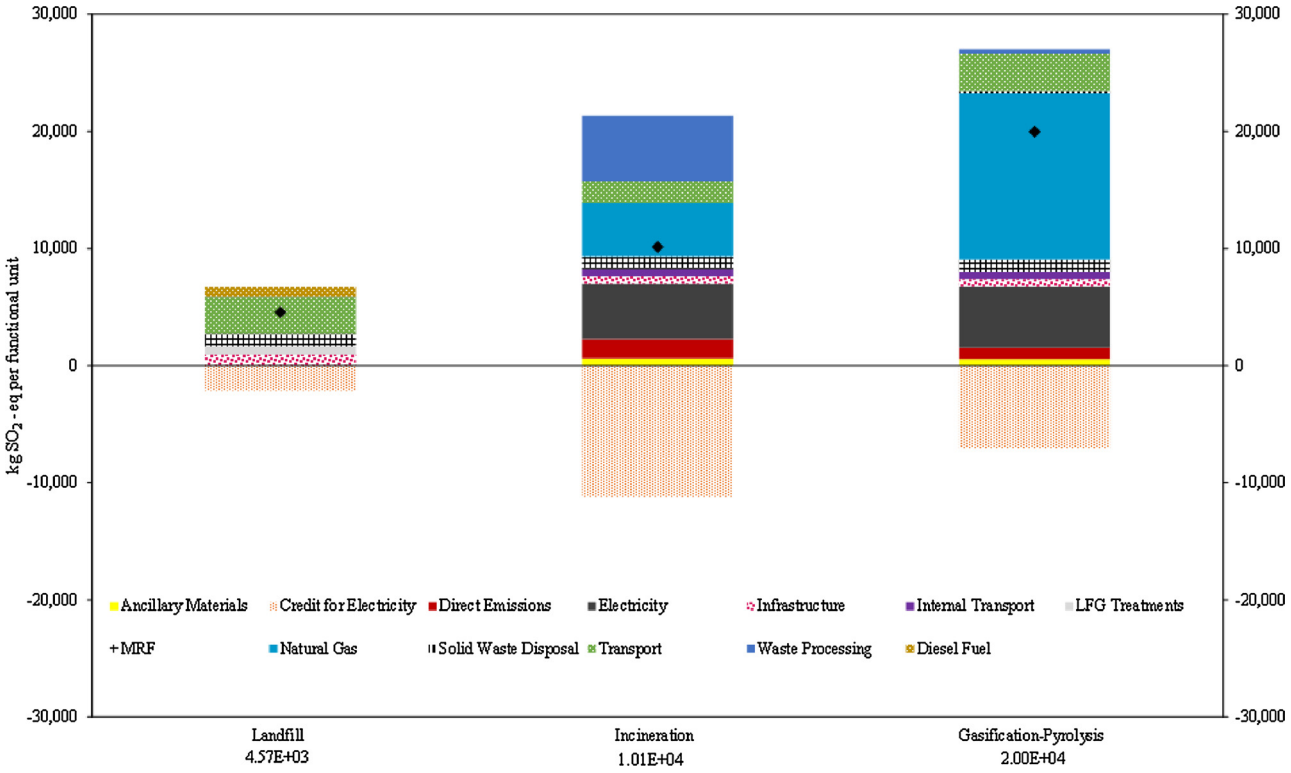

Fig. 2. Acidification potential impact of the functional unit and processes under each scenario. The black diamond represents the net outcome, with a positive number being an environmental burden and a negative number being an environmental benefit. The results show that landfill contributed to the lowest acidifying impact.

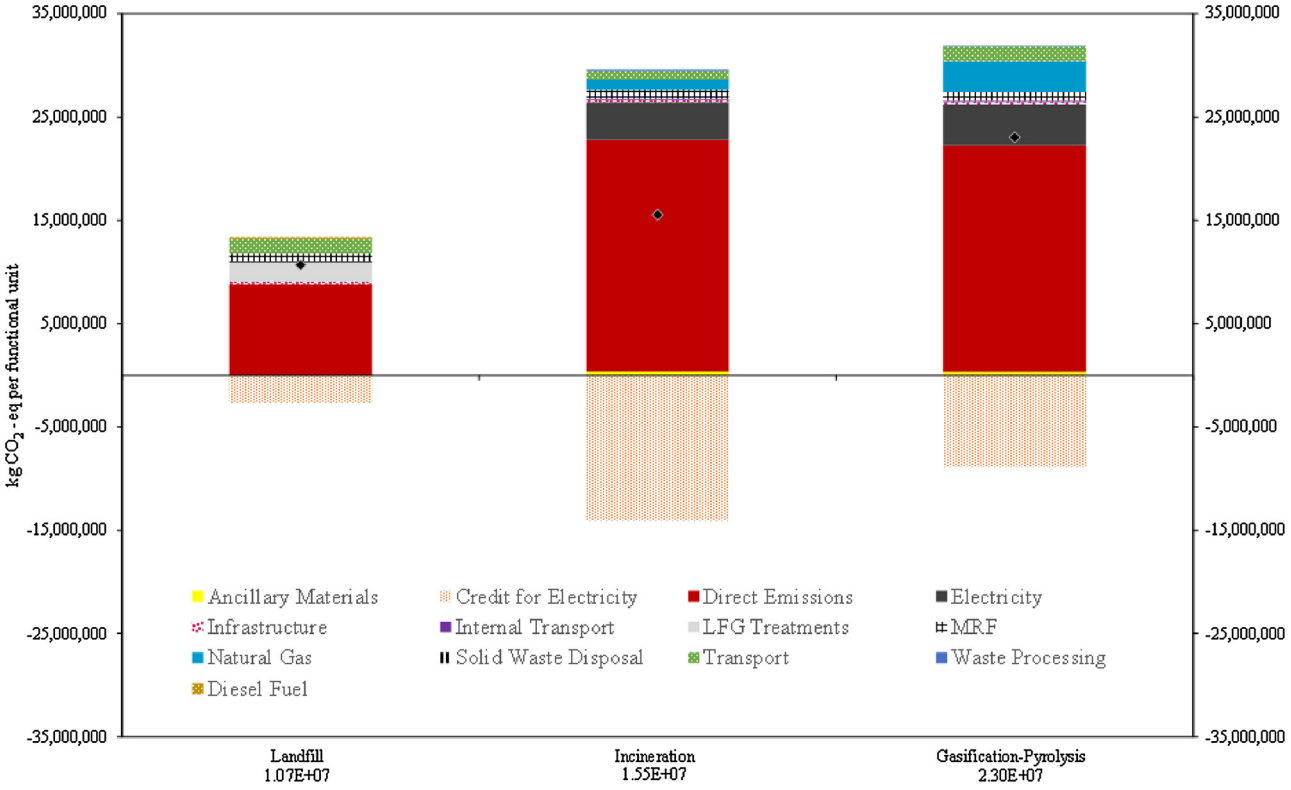

Fig. 3. Climate change potential impact of the functional unit and processes under each scenario. The black diamond represents the net outcome, with a positive number being an environmental burden and a negative number being an environmental benefit.

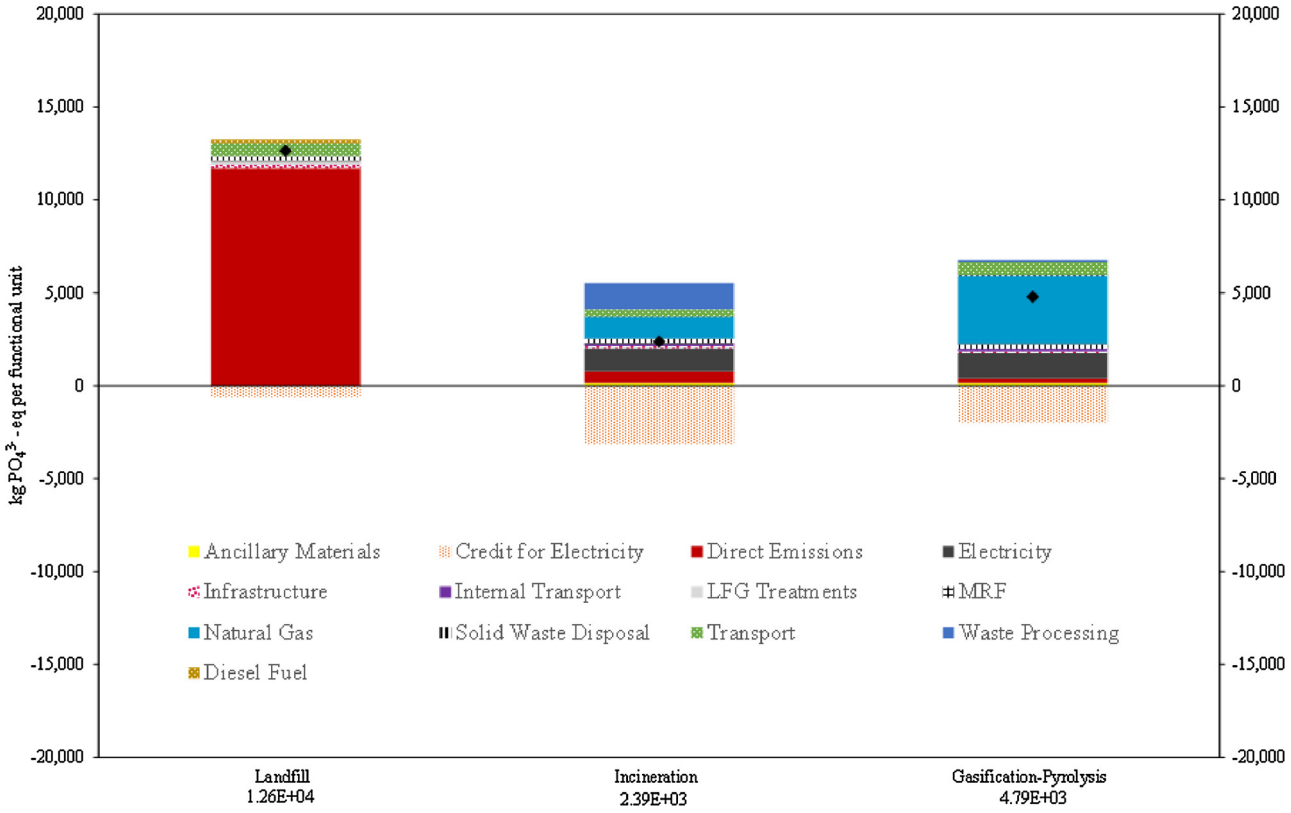

Fig. 4. Eutrophication potential impact of the functional unit and processes under each scenario. The black diamond represents the net outcome, with a positive number being an environmental burden and a negative number being an environmental benefit.

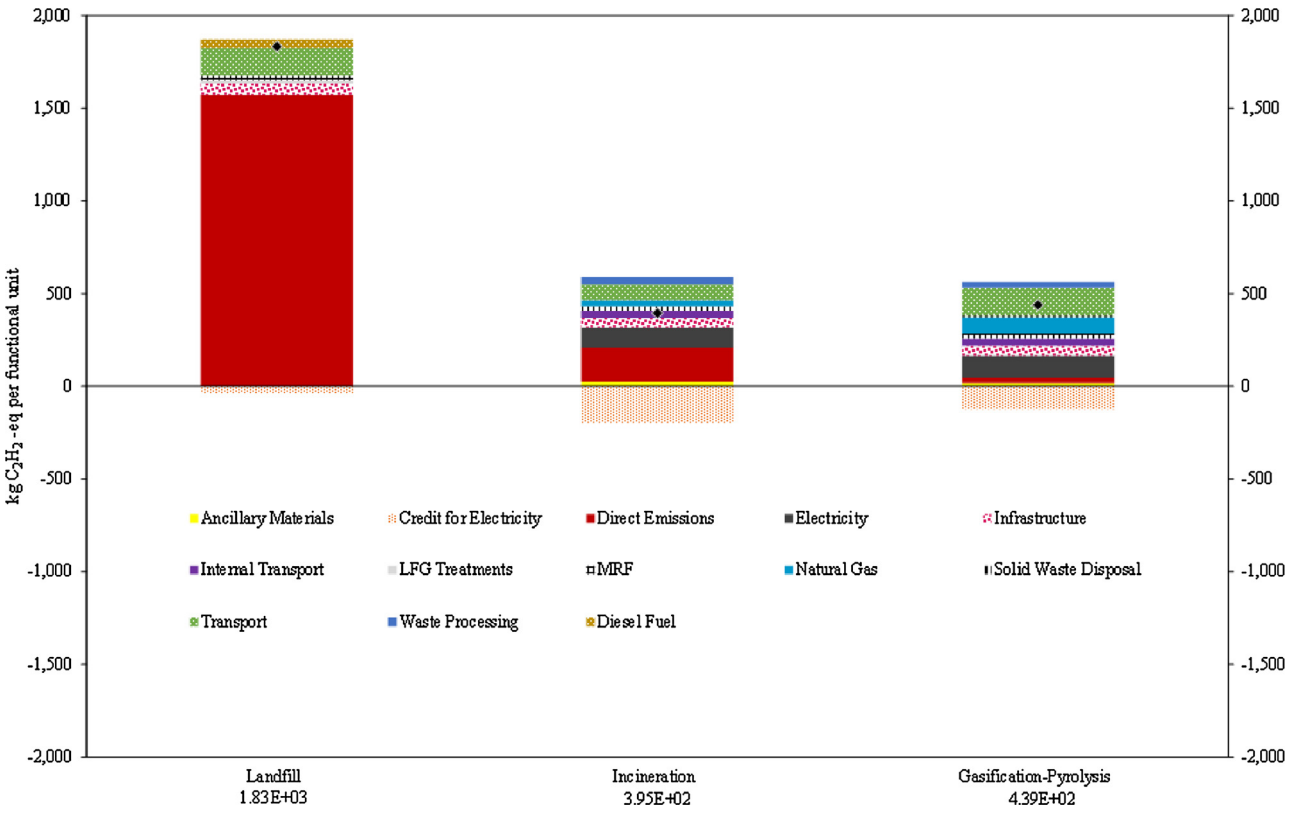

Fig. 5. Photochemical oxidation potential impact of the functional unit and processes under each scenario. The black diamond represents the net outcome, with a positive number being an environmental burden and a negative number being an environmental benefit.

Table 10
Characterised acidifying substance emissions for AP at 5% cut-off for the functional unit under each scenario (GP = gasification-pyrolysis).

Flows	Unit	Landfill	Incineration	GP
Total of all compartments	kg SO_2 –eq	4.57×10^3	1.01×10^4	2.00×10^4
Remaining substances	kg SO_2 –eq	4.31×10^1	5.65×10^2	1.56×10^2
Nitrogen oxides	kg SO_2 –eq	3.45×10^3	8.30×10^3	1.78×10^4
Sulfur dioxide	kg SO_2 –eq	1.08×10^3	1.28×10^3	2.06×10^3

Table 11
Characterised greenhouse gas substance emissions for CCP at 5% cut-off for the functional unit under each scenario (GP = gasification-pyrolysis).

Flows	Unit	Landfill	Incineration	GP
Total of all compartments	kg CO_2–eq	1.07×10^7	1.55×10^7	2.30×10^7
Remaining substances	kg CO_2–eq	5.86×10^4	3.83×10^5	2.88×10^5
Carbon dioxide, biogenic	kg CO_2 –eq	4.07×10^6	9.35×10^6	9.02×10^6
Carbon dioxide, fossil	kg CO_2–eq	–	5.81×10^6	1.37×10^7
Methane, biogenic	kg CO_2–eq	6.54×10^6	–	–

CO_2 emissions.

Table 11 presents the elementary flows of GHG emissions contributing to CCP for each system. In landfill, biogenic CO_2 contributes to 37% and biogenic CH_4 contributes to 61% of the total impact mostly from the degradation of waste. In incineration, biogenic CO_2 contributes to 59% and fossil CO_2 contributes to 38% of the total impact. This is mostly driven by direct emissions from the combustion of waste, and electricity usage from the NSW grid also contributes to emissions of fossil CO_2. In gasification-pyrolysis, biogenic CO_2 contributes to 39% and fossil CO_2 contributes to 59% of the total impact. GHG emissions are primarily generated from the combustion of waste, and electricity and natural gas consumption provide addition fossil CO_2 emissions.

For EP, landfill has the highest net impact at 1.27×10^4 kg PO_4^{-3}-eq, 372% greater than incineration and 166% greater than gasification-pyrolysis. The direct emissions of nitrate from the leachate, after processing through the wastewater treatment plant are the primary driver of EP.

Table 12 presents the elementary flows of euthrophying substances contributing to EP for each system. In landfill, nitrate emissions contribute to 87% and nitrogen oxides contribute to 7% of the total impact. The nitrate emissions are from the leachate and nitrogen oxides are mostly from combustion of diesel including in diesel landfill operations, MRF operations and road transport fuel. In incineration, nitrogen oxides contribute to 91% and nitrate emissions contribute to 9% of the total impact. Nitrogen oxides are mostly from the combustion of waste, electricity usage from the NSW grid and the combustion of natural gas. In gasification-pyrolysis, nitrogen oxides contribute to 97% of the total impact mostly from the combustion of natural gas.

Table 12
Characterised euthrophying substance emissions for EP at 5% cut-off for the functional unit under each scenario (GP = gasification-pyrolysis).

Flows	Unit	Landfill	Incineration	GP
Total of all compartments	kg PO_4^{3-}–eq	1.26×10^4	2.39×10^3	4.79×10^3
Remaining substances	kg PO_4^{3-}–eq	7.07×10^2	2.29×10^2	1.66×10^2
Nitrate	kg PO_4^{3-}–eq	1.10×10^4	–	–
Nitrogen oxides	kg PO_4^{3-}–eq	8.97×10^2	2.16×10^3	4.62×10^3

Finally, the POP has the highest net impact for landfill at 1.83×10^3 kg C_2H_2 –eq, 357% greater than incineration and 323% greater than gasification-pyrolysis. The degradation of MRF residual waste in landfill to biogenic CH_4 is the primary cause of POP, followed by CO emissions from transport.

Table 13 presents the elementary flows of smog producing substances contributing to POP for each system. In landfill, CH_4 emissions contribute 86% and CO contributes to 7% of the total impact. CH_4 emissions are from the degradation of waste to air. In incineration, biogenic CO contributes to 34% and fossil CO contributes to 30% of the total impact. Both biogenic and fossil CO are from the combustion of waste and the NSW electricity grid. In gasification-pyrolysis, total CO emissions contribute to 55%, hexane contributes to 6%, CH_4 contributes to 12% and SO_2 contributes to 17% of the total impact.

4.4. Treatment of biogenic carbon dioxide

The results of applying the biogenic CO_2 non-neutral method (baseline) and the biogenic CO_2 neutral method are reported in Fig. 6. The sensitivity case (S) assumes that biogenic CO_2 air emissions are greenhouse neutral and counts carbon in landfill as a sequestration credit. Compared with the baseline results, the biogenic CO_2 neutral method gives CCP results that are lower by 1.18×10^7 kg CO_2 -eq in landfill, 9.36×10^6 kg CO_2 -eq in incineration and 9.04×10^6 kg CO_2 -eq in gasification-pyrolysis. Regardless of the accounting method, the landfill has the lowest CCP.

4.5. Uncertainty analysis

The uncertainty results presented in Table 14 show that when comparing POP for treatment of MRF residual waste in landfill to incineration, and landfill to gasification-pyrolysis, thermal treatments outperform landfill with 100% certainty (i.e. 1000 of 1000 runs). Conversely, when comparing AP is landfill outperforms incineration with 99.8% certainty, and gasification-pyrolysis with 100% certainty. Incineration outperforms gasification with 15.8% certainty for CCP.

4.6. Sensitivity analysis – MRF residual waste composition

The results of the sensitivity analysis of MRF residual waste composition are presented in Tables 15–18. The directional outcomes are consistent with gasification-pyrolysis having the highest AP and CCP, regardless of waste composition. The directional outcomes are also consistent with landfill having the highest EP and POP. However, Waste Composition 3 based on a waste composition in Enviros Consulting (2009), shows that for AP, incineration has the lowest impact, rather than landfill, due to an offset against the NSW electricity grid. Waste Composition 2 based on a waste composition in Assamoi and Lawryshyn (2012) has the lowest CCP for incineration, rather than landfill.

5. Discussion

Normalization by the totally yearly environmental load by region is a common approach to understand the magnitude of characterized impacts (Clift, Doig & Finnveden 2000). The normalized results in our study are characteristically Australian and temporal to 2015. The weighting method of aggregating multiple environmental impacts to a single score is an area of debate. Weighting as a means of LCA interpretation can be considered controversial because it inherently requires reasoning based on social, political or ethical values to determine the importance of different indicators (Clift, Doig & Finnveden 2000). The three different approaches for weighting can are: monetary methods

Table 13
Characterised photochemical oxidating substance emissions for POP at 5% cut-off for the functional unit under each scenario (GP = gasification-pyrolysis).

Flows	Unit	Landfill	Incineration	GP
Total of all compartments	kg C_2H_2 -eq	1.83×10^3	3.95×10^2	4.39×10^2
Remaining substances	kg C_2H_2 -eq	1.42×10^2	4.25×10^1	3.62×10^1
Carbon monoxide	kg C_2H_2 -eq	1.20×10^2	–	1.46×10^2
Carbon monoxide, biogenic	kg C_2H_2 -eq	–	1.37×10^2	4.58×10^1
Carbon monoxide, fossil	kg C_2H_2 -eq	–	1.19×10^2	5.08×10^1
Hexane	kg C_2H_2 -eq	–	–	2.51×10^1
Methane	kg C_2H_2 -eq	1.57×10^3	4.47×10^1	5.23×10^1
Sulfur dioxide	kg C_2H_2 -eq	–	5.10×10^1	8.25×10^1

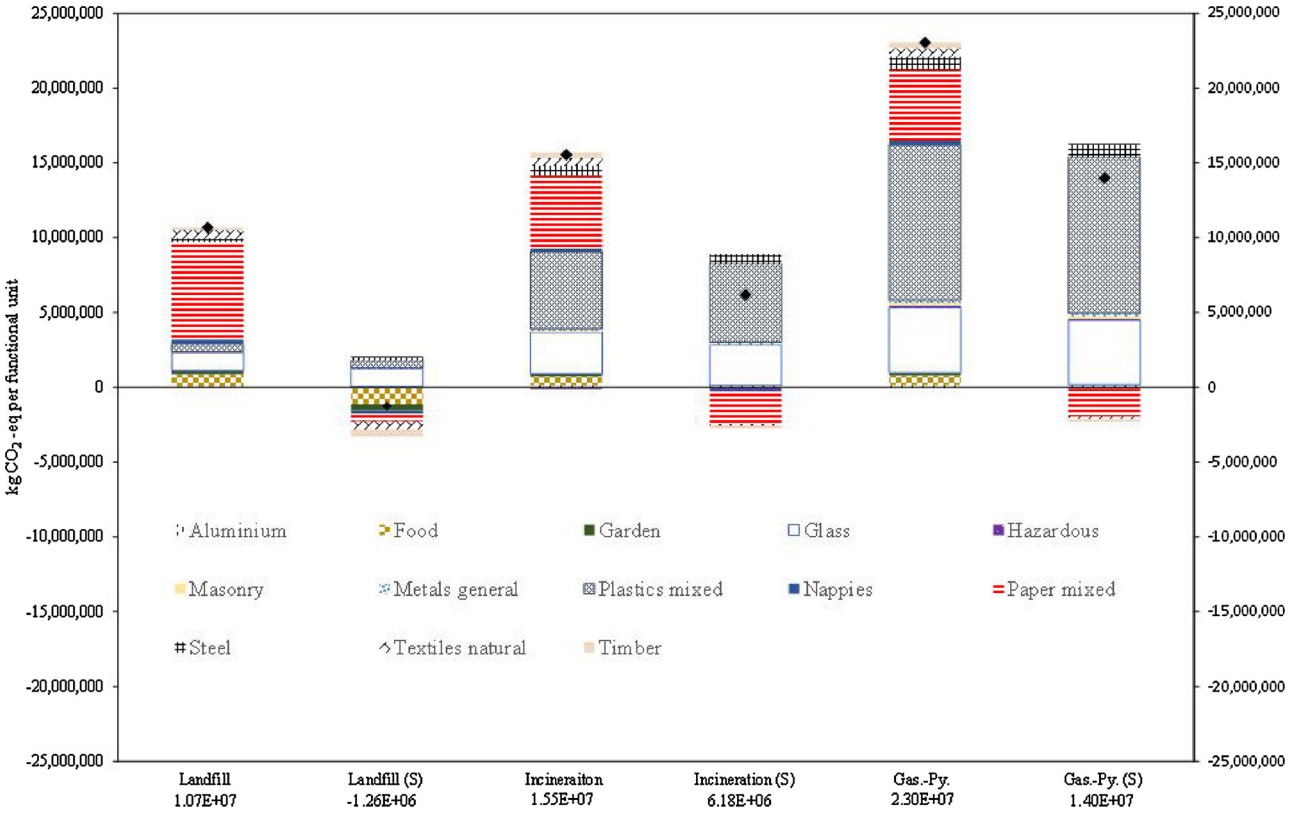

Fig. 6. Disaggregated results for climate change potential of the functional unit under landfill, incineration and gasification-pyrolysis. The black diamond represents the net result for each scenario.

Table 14
Results of comparative Monte Carlo simulation for the treatment of 30,000 t of MRF residual waste showing the number of occurrences (%) where a system was greater than the other (GP = gasification-pyrolysis).

Indicator	MC runs where landfill > incineration	MC runs where landfill > GP	MC runs where incineration > GP
AP	0.20%	0%	5.10%
CCP	6.70%	0.70%	15.80%
EP	99.20%	97.40%	66.60%
POP	100%	100%	31.30%

Table 15
AP results of sensitivity analysis testing the composition of different MRF residual waste mixes based on other studies.

Waste	Unit	Landfill	Incineration	Gasification-Pyrolysis
Baseline	kg SO_2 –eq	4.57×10^3	1.01×10^4	2.00×10^4
Waste Composition 1	kg SO_2 –eq	4.40×10^3	4.29×10^3	1.75×10^4
Waste Composition 2	kg SO_2 –eq	1.63×10^3	6.10×10^3	1.43×10^4
Waste Composition 3	kg SO_2 –eq	3.18×10^3	-2.34×10^3	1.16×10^4

(i.e. willingness to pay for environmental damages), sustainability and target methods, and panel approach (Khoo, 2009). The bias of this study is towards the Australian weighting method, BPIC (Bengtsson, J, Howard & Kneppers 2010); however, the BPIC method, published in 2010, was based on a series of surveys across Australia, questioning stakeholders' perceived importance of different environmental impacts, consistent with the panel approach. Regardless of the weighting method applied, landfill still had the lowest single score result. However, the final score may be sensitive to exclusion of other indicators not analysed in our study, including terrestrial eco-toxicity, and marine and freshwater eco-toxicity. Another consideration of the final score based on the Australian BPIC method are changes in value judgements of the importance of different environmental impacts since its publication 2010.

Waste and energy companies have claimed that MRF residual waste is better managed by incineration than landfill. For example, Energy

Table 16
CCP results of sensitivity analysis testing the composition of different MRF residual waste mixes based on other studies.

Waste	Unit	Landfill	Incineration	Gasification-Pyrolysis
Baseline	kg CO_2 –eq	1.07×10^7	1.55×10^7	2.30×10^7
Waste Composition 1	kg CO_2 –eq	1.06×10^7	1.93×10^7	3.27×10^7
Waste Composition 2	kg CO_2 –eq	2.34×10^7	2.31×10^7	2.86×10^7
Waste Composition 3	kg CO_2 –eq	1.92×10^7	2.72×10^7	4.10×10^7

Table 17
EP results of sensitivity analysis testing the composition of different MRF residual waste mixes based on other studies.

Waste	Unit	Landfill	Incineration	Gasification-Pyrolysis
Baseline	kg PO_4^{3-}eq	1.26×10^4	2.39×10^3	4.79×10^3
Waste Composition 1	kg PO_4^{3-}eq	1.87×10^4	8.64×10^2	4.05×10^3
Waste Composition 2	kg PO_4^{3-}eq	1.63×10^4	1.28×10^3	3.38×10^3
Waste Composition 3	kg PO_4^{3-}–eq	2.08×10^4	-1.04×10^3	2.45×10^3

Table 18
POP results of sensitivity analysis testing the composition of different MRF residual waste mixes based on other studies.

Waste	Unit	Landfill	Incineration	Gasification-Pyrolysis
Baseline	kg C_2H_2 –eq	1.83×10^3	3.95×10^2	4.39×10^2
Waste Composition 1	kg C_2H_2 –eq	1.83×10^3	2.93×10^2	4.11×10^2
Waste Composition 2	kg C_2H_2 –eq	4.31×10^3	3.19×10^2	3.43×10^2
Waste Composition 3	kg C_2H_2 –eq	3.49×10^3	1.81×10^2	2.69×10^2

Australia has claimed that the management of 100,000 t of RDF by waste-to-energy technology in NSW can avoid 60,000 t of net GHG emissions (Energy Australia, 2017). However, the results from our study show that the treatment of the MRF residual waste through waste-to-energy technology did not outperform landfill for characterized GHG emissions. Indeed, both the baseline and sensitivity analysis results show incineration and gasification-pyrolysis are net contributors of GHG, irrespective of the approach taken to assess biogenic CO_2. However, there are a number of unknown key factors in the Energy Australia (2017) case to compare directly to our study. Firstly, in our study, thermal credits for incineration were excluded on the basis that there was no market for as a co-product like there was for electricity, which is easily exported to the grid. The inclusion of heat as a co-product would have provided incineration with a lower net result for GHG emissions, reducing CCP impact by 19% to 1.28×10^7 kg CO_2 -eq per functional unit (30,000 t of MRF residual waste), using an efficiency for thermal heat recovery of 28.5% (Doka, 2013). Secondly, the estimations for the Mount Piper waste-to-energy plant may have used a higher conversion efficiency factor from heat to electricity generation than our study which used 15.8% for incineration estimated from Doka (2013). In our study, Waste Composition 3, based on MRF residual waste from Enviros Consulting (2009), had the highest electricity credit potential, 0.89 MW h per kg, driven by a composition fraction of mixed plastic at 46.8% (LHV of 30.8 MJ per kg) and a fraction of mixed paper at 37.9% (LHV of 14.1 MJ per kg). This is still 19% lower than that the Mount Piper waste-to-energy plant, that estimates an electricity production of 1.1 MW h per tonne of RDF (Luger, 2017). Better insight into estimated compositions of residual waste is essential for knowing potential environmental impacts of waste-to-energy in Sydney, and proposed controls to ensure the heating values of waste feedstock are consistent over time.

The conclusions from Assamoi and Lawryshyn (2012) were sensitive to credits from avoided electricity production. Incineration of the MRF residual waste had the lowest impact in each category; AP, GWP (characterized GHG emissions) and nutrient enrichment (characterized nitrate equivalent emissions). Without electricity credits, Assamoi and Lawryshyn (2012) found that landfill had the lowest impact in each category. By contrast, our study showed incineration and gasification-pyrolysis systems had lower EP but the landfill system had lower AP and CCP. When analysing the composition of MRF residual waste based on Assamoi and Lawryshyn (2012) in our model (Waste Composition 2), similarly, the results found that incineration had the highest AP. However, the CCP from incineration was 1.3% lower than landfill system with both having net GHG burdens with the inclusion of electricity credits. The model in the Assamoi and Lawryshyn (2012) LCA used offsets from electricity production from four coal-fired, one natural-gas-fired and one oil-fired power plant; and our study had 9.2% of the electricity mix from renewables, using the NSW electricity grid in 2013/2014 (Commonwealth Of Australia et al., 2015). Also, the Assamoi and Lawryshyn (2012) model excluded biogenic CO_2 emissions from the combustion of waste. In our study, the CCP results found that biogenic CO_2 neutral method, compared to the non-neutral method reduce the incineration impact by 59% relative to the baseline waste composition.

The appropriateness of GHG accounting methods that exclude biogenic CO_2 from the scope as is described by the IPCC is a debate that is not yet resolved (Steubing, 2015; Claflin, 2011; Stashwick, 2015;Brandão et al., 2012). The biogenic carbon neutral method assumes all biogenic CO_2 emissions are removed by photosynthesis from plants, however, this assumption comes with uncertainty due to the dynamic nature of the carbon stock in forests (Helin et al., 2013). The method relies on separate accounting of net carbon stocks through land-use changes over time (Claflin, 2011). Furthermore, this method ignores the time required to re-sequester combusted biomass. Therefore, although the method correctly assumes that atmospheric impacts of biogenic CO_2 and fossil CO_2 are equivalent (Steubing, 2015), it might underestimate GHG impacts. Brandão et al. (2012) discusses several methods to quantify biogenic CO_2 in LCIA with consideration of the actual time required to re-sequester combusted biomass into a reference system. However, regardless of these arguments, a change in CO_2 accounting methodology had no effect on the directional outcomes of the systems considered here.

The Zaman (2010) study used LCA to measure the environmental performance of one tonne of MSW under landfill, incineration and gasification-pyrolysis waste treatment scenarios in Sweden. In relation to the performance of gasification-pyrolysis, Zaman (2010) found that incineration had a 232% greater AP impact than gasification-pyrolysis. However, the results of our study disagree with that finding, showing gasification-pyrolysis had a 75.9% greater AP impact than incineration. The driver for greater AP from gasification-pyrolysis is additional natural gas input required to maintain higher process temperatures; 1.9 MJ per kg for gasification-pyrolysis and 0.61 MJ per kg for incineration. Although the natural gas input in Zaman (2010) is not reported, the start-up energy required for gasification-pyrolysis is 436% greater than for incineration and the energy generated throughout the process is 126% greater, thereby creating a greater net credit by gasification-pyrolysis from the electricity grid in Sweden. In our study, the uncertainty analysis comparing each system found that the comparison of incineration and gasification-pyrolysis had the least conclusive

outcomes. In comparative MC simulations, a value of 90% certainty may be used to confidently determine that one system outperforms another (Golsteijn 2015). The uncertainty factors were derived from qualitative analysis of six characteristics. The collection of empirical data from waste-to-energy plants in Australia can be expected to increase confidence of future LCA studies.

The results from our study are important because they show relevance of independently verified research to support public policy and planning. The limitations of this study include credits from the NSW electricity grid based on production inputs in the 2013/2014 period. Policy advice on energy to the federal government is expected to lead the national electricity market towards lower grid GHG emissions over the coming decades (Commonwealth of Australia, 2016a). Therefore, credits from avoided electricity may have lower effect on the net climate change impact. Finally, the practical issue of diminishing landfill capacity in metropolitan cities of Australia remains an important consideration, especially if landfilling at future sites requires longer transport distances.

6. Conclusions

This study aimed to calculate the environmental performance of the MRF residual waste in New South Wales (NSW) under three waste treatment scenarios in response to a high level of interest in waste-to-energy technology by Australian state governments and industry. The study developed three end-of-life waste management scenarios for landfill, incineration and gasification and used an LCA model to measure and compare environmental impacts. The limitations of this study include the geographical scope being limited to NSW and technological co-relation to the NSW electricity grid. The conclusions from this study are that the MRF residual waste is best managed by:

- Landfilling to minimise general environmental impact on a single score, weighted and normalized to Australian emissions references
- Landfilling to minimise acidifying emissions
- Landfilling to minimise greenhouse gas emissions, regardless of the GHG accounting method used (i.e. biogenic CO_2 non-neutral/neutral methods)
- Incineration to minimise euthrophying emissions
- Gasification-pyrolysis to minimise photochemical oxidation emissions

Future work could include a consequential LCA (rather than attributional LCA used in this study) that would address estimated changes in Sydney's MRF residual waste as a consequence of market demands for consumer goods, available capacity in landfill, odour control at landfill and regional expansion of scope to manage waste.

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